DISCUSSION PAPER

January 2012 ■ RFF DP 12-01

# Conservation Return on Investment Analysis

## *A Review of Results, Methods, and New Directions*

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RESOURCES FOR THE FUTURE

# Conservation Return on Investment Analysis: A Review of Results, Methods, and New Directions

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## Abstract

Conservation investments are increasingly evaluated on the basis of their return on investment (ROI). Conservation ROI analysis quantitatively measures the costs, benefits, and risks of investments so conservancies can rank or prioritize them. This paper surveys the existing conservation ROI and related literatures. We organize our synthesis around the way studies treat recurring, core elements of ROI, as a guide for practitioners and consumers of future ROI analyses. ROI analyses involve quantification of a consistent set of elements, including the definition and measurement of the conservation objective as well as identification of the relevant baselines, the type of conservation investments evaluated, and investment costs. We document the state of the art, note some open questions, and provide suggestions for future improvements in data and methods. We also describe ways ROI analysis can be extended to a broader suite of conservation outcomes than biodiversity conservation, which is the typical focus.

**Key Words:** return on investment, conservation planning, reserve site selection

**JEL Classification Numbers:** Q20, Q30, Q51, Q57

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# Conservation Return on Investment Analysis: A Review of Results, Methods, and New Directions

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## 1. Introduction

Conservation organizations face important choices as they protect, restore, and manage natural resources. What are the most important conservation targets? How can we get the most conservation for a given budget? One way these questions can be addressed is by return on investment (ROI) analysis. Conservation ROI analysis quantitatively measures the costs, benefits, and risks of investments so conservancies can rank or prioritize them. It also can be used after an investment is made to evaluate its success or failure.

ROI analysis is as applicable and relevant to conservation investments as it is to corporate, other private, or public investments, where it is applied routinely (Brealey et al. 2006). While more difficult to express in dollars and cents, conservation outcomes are socially valuable and thus akin to "returns" relevant to setting strategy and evaluating success.

This paper surveys the existing conservation ROI and related literatures. Our goals are to summarize the state of the art in terms of results, data, and methods and then identify strategies to improve the scope, accuracy, and applicability of conservation ROI analysis. The paper also describes the ways in which ROI analysis can be extended to a broader suite of conservation outcomes than biodiversity conservation, which is the typical focus. In particular, we discuss ROI's application to the provision of ecosystem goods and services.

Conservation ROI analysis requires measurement or prediction of biophysical changes arising from conservation and (at a minimum) economic assessment of conservation's costs. Economists and conservation scientists are collaborating on analyses that suggest ROI analysis is both achievable and important to conservation strategy. As will be seen in this review, the simple inclusion of costs as a factor in conservation planning can yield enormous conservation benefits. Studies demonstrate that if adopted, ROI-based planning would in many cases significantly alter

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the location and targets of conservation, lead to more protection and higher-quality conservation outcomes, and save significant amounts of money (Balmford et al. 2000; Moore et al. 2004; Murdoch et al. 2007, 2010).[1] Further development of data and models to capture ecological threats, ecological benefits beyond biodiversity, and social outcomes arising from conservation could have correspondingly important implications.

## 2. What Is Conservation ROI Analysis?

ROI methods are derived from, and consistent with, economic principles that emphasize the quantification of costs and benefits, discounting of future costs and benefits, and adjustments for risk. Among the wider range of approaches to conservation assessment (Salafsky and Margoulis 1998), ROI can be thought of as the most economic form of evaluation. But despite its economic features, ROI analysis does not necessarily measure monetary or anthropocentric outcomes. It can instead focus on the way to achieve a purely biophysical outcome at least cost. In fact, this is the most common form of conservation ROI analysis conducted to date.

All ROI analyses compare an investment’s costs to some measure of benefits. Costs may be one-time and upfront or recurring over time. Benefits are depicted as changes in desired outcomes attributable to the investment. In a business setting, new revenues attributable to the investment are the conventional benefit measure. In conservation, benefit measures include biodiversity improvements, land area conserved, or other desirable biophysical and social outcomes.

In all cases, an investment’s benefits are judged in relation to the baseline conditions prevailing if the investment is not made. Accordingly, ROI analysis requires not only prediction or measurement of the investment’s outcomes but also measurement or prediction of baseline conditions.

While an oversimplification, conservation investments in restoration, enhancement, or resource management (e.g., grazing restrictions) tend to produce improvements in current conditions. In contrast, land conservation and protection tend to preserve current conditions but avoid or slow future ecological losses. Both can generate positive conservation investment

[1] The ROI benefits demonstrated by these studies arise from the inclusion of costs as a project-selection factor. Cheaper projects do not necessarily select the highest-value conservation targets in ecological terms, but they allow acquisition of a larger set of protections that yield more conservation benefit in aggregate.

returns. Note that evidence of biophysical improvements (e.g., improvements in biodiversity) is not necessary to demonstrate a positive conservation ROI. When baseline conditions are otherwise in decline, maintenance of existing conditions produces a conservation benefit. As will be clear from this survey, measurement and prediction of baseline ecological conditions is a central challenge facing conservation ROI analysis.

In the conservation literature, ROI most commonly refers to ex ante analysis of investment portfolios designed to achieve a single objective—usually a biodiversity outcome. In general, this form of ROI analysis combines data on biodiversity richness and threats with data on the costs of land acquisition or protection. Single-objective ROI analysis is largely synonymous with cost-effectiveness analysis, where the goal is to achieve a given outcome at least cost or spend a given amount of money to achieve the greatest possible benefit. The goal of such analyses is the prioritization of lands for conservation across regions or countries. While this literature is our focus, we emphasize that multiobjective and social-outcome ROI analysis is of increasing interest within the conservation community. We turn to this issue in Section 5.

We proceed by briefly reviewing conservation planning approaches based on purely biophysical criteria. We then present a detailed review of the growing ROI literature that includes conservation costs as a decision factor. Next we discuss multiobjective ROI analysis, including analysis of ecosystem service-based returns to conservation. This literature is much more limited. However, we review developments in a set of related literatures on ecosystem services and environmental valuation pertinent to the future development of multiobjective ROI tools. The paper concludes with thoughts on the practical implementation of ROI analysis by conservation planners.

## 3. A Brief History of Conservation Planning and Objectives

Systematic conservation planning has a long history, including early efforts in Europe and the United States to identify public lands for protection and acquisition (Fox 1986) and efforts to achieve soil conservation in the early 20$^{th}$ century (Harlow 1994). The application of ecological science to conservation planning gained momentum with the creation of nongovernmental conservancies and environmental advocates, such as the Nature Conservancy (TNC) and Defenders of Wildlife, in the latter half of century. In the late 1980s the U.S. Fish and Wildlife Service launched a national resource survey to identify gaps in the network of conserved lands. These kinds of biodiversity assessments continue today within the United States and around the world, including surveys by the U.S. Geological Survey and nonprofit organizations like NatureServe.

An outgrowth of these surveys are conservation plans designed to address biodiversity "gaps" by identifying specific lands for conservation action and acquisition. Examples include U.S. State Wildlife Action Plans, TNC's ecoregional plans (Groves et al. 2000), Conservation International's biodiversity hotspot designations (Myers et al. 2000), and dozens of other governmental, nongovernmental, and academic prescriptions (e.g., Cowling et al. 2003).

Some three decades ago, the development of the first reserve site selection (RSS) algorithms commenced a major shift toward the use of quantitative methods in conservation priority setting (Williams et al. 2004). Previously, potential target areas typically were assessed and ranked based on expert assessments, often using some form of scoring approach (Margules and Usher 1981; Smith and Theberge 1986; Pressey 2002). In an important extension to the scoring approach, Jamie Kirkpatrick (1983) developed an iterative procedure to repeatedly reassess candidate sites based on how their species compare to those already preserved by the existing reserves. After identifying the first priority area (with the highest initial score), each candidate site is reassessed. Its score is reduced if the site includes species already preserved, while sites with species not yet preserved have their scores upgraded. The process is then repeated in the selection of further priority sites. Kirkpatrick's assessment of conservation priorities for rare plants in Tasmania is considered the first RSS algorithm (Pressey 2002).

Independent of Kirkpatrick (1983), other researchers were proposing similar ideas in the early to mid-eighties. Ackery and Vane-Wright (1984) evaluated options for critical butterfly conservation areas. For example, they proposed using an RSS-like algorithm to identify a set of areas to represent all known species (Vane-Wright et al. 1991). Margules and Nicholls (1987) used probabilistic estimates of species occurrence to examine the preservation of specific plant communities in South Australia, developing an algorithm intended to identify a set of conservation areas to achieve a minimum probability of representing different communities. Pressey (2002) vividly accounts these early developments and discusses several other studies that helped motivate RSS approaches as a central tool in conservation prioritization.

As conservation planning has grown in sophistication, it has moved toward rigorous mathematical optimization (Possingham et al. 2000). Another central development involves expanding the relatively simple characterizations of biodiversity to more realistic considerations of ecological, population, and evolutionary processes and the spatial configurations of protected lands on which biodiversity depends (Groves et al. 2002). Increasingly, assessments of climate change, invasive species, and other threats to biodiversity are included in conservation planning (McLeod et al. 2009). The number of academic publications using RSS methods has vastly increased in the last decade or so, and these methods are somewhat routinely used to guide practical conservation decisions. Several customized software are available to provide conservation decision support, including MARXAN (Watts et al. 2009).

Although the boundaries and realism of RSS methods have considerably expanded in the last decade or so, incorporating increasingly realistic representations of the temporal and spatial dynamics and interdependencies of ecosystem and landscape processes remains one of the key challenges (Williams et al. 2004).

## 4. Conservation Planning with Costs as a Factor

While conservation is inevitably constrained by economic realities, costs as a planning factor mostly were absent from systematic conservation planning until the late 1990s. But since then, inclusion of costs as a planning factor has become a central concern in conservation planning (Ando et al. 1998; Polasky et al. 2001; Naidoo et al. 2006). Many studies have demonstrated that planning based on ecological benefits produced per dollar spent—the essence of ROI analysis—can help the same conservation budget achieve considerably greater conservation benefits (Naidoo et al. 2006).

Conservation costs were first accounted for in single-species conservation problems, such as work by Haight (1995), Montgomery et al. (1994), Marshall et al. (1998), and Hof and Bevers (1998). Ando et al. (1998) showed that significantly greater biodiversity could be conserved and biodiversity targets could be achieved at much lower costs by explicitly accounting for costs in the planning process. They found that biodiversity targets could be met at 25–50 percent of the costs of plans that only considered spatial heterogeneity of biodiversity. The study by Ando et al., in particular, documented the importance of spatial heterogeneity in conservation costs as a driver of ROI analysis' benefits as a planning tool. Significant cost variation across conservation portfolios has been demonstrated by numerous subsequent studies (Balmford et al. 2003; Ferraro 2003; Polasky et al. 2001).

Examples of the conservation benefits of cost-based planning include a 66 percent gain in vertebrates species coverage in African conservation when costs were included (Moore et al. 2004). At the global level, Balmford et al. (2000) found that twice as many mammal species could be conserved for the same budget when costs were considered. Examining vertebrate conservation in Oregon, Polasky et al. (2001) found that a budget-constrained solution cost less than 10 percent of a solution that ignored costs. Accounting for potential reductions in lobster catch during marine reserve planning for southern Australia reduced costs by more than 30 percent (Stewart and Possingham 2005). An analysis of riparian buffer acquisition found that only 16 percent of the benefits obtained by considering costs and benefits together could be gained by considering only biophysical characteristics (Ferraro 2003). Naidoo et al. (2006) summarizes this earlier ROI literature.

Since 2006 the conservation ROI literature has grown and provided further evidence that costs are important as a conservation-planning factor. For example, Murdoch et al. (2007), Naidoo and Iwamura (2007), Low et al. (2010), and Provencher et al. (2010) documented ROI analysis' ability to dramatically change site prioritizations and yield better and cheaper conservation outcomes. Naidoo and Iwamura (2007) found that plans selecting global priority bioregions based on cost-effectiveness rather than benefits alone protected endemic vertebrate species at much lower cost. Specifically, conserving one-quarter or two-thirds of all endemic species could be achieved at less than 10 percent and 25 percent of the costs, respectively. When comparing conservation ROI targeting to two existing priority schemes, they also found that the same number of endemic vertebrates could be represented at 27 percent of the opportunity costs and the same number of threatened endemic species could be conserved for just 12 percent of costs.

Joseph et al. (2009) showed that accounting for both costs and likelihood of project success substantially increased the number of species managed. Carwardine et al. (2008a) found that failing to consider socioeconomic factors resulted in reserve design that targeted twice as much cropped land and incurred 1.5 times the opportunity cost. In a parallel study, Carwardine et al. (2008b) also found that using the cost surrogate that most closely reflects the planned conservation action can cut the cost of achieving biodiversity goals by half; using area as a surrogate for cost was 1.4–2.3 times more expensive. Murdoch et al. (2010) found that targeting only benefits conserved twice the benefit as ROI but at 50 times the cost, whereas under a budget constraint ROI conserved three times the benefit. Similarly, Underwood et al. (2008) found that ROI protected between 32 percent and 69 percent more species compared to the other priority-setting approaches. In a retrospective analysis of conservation in California, Underwood et al. (2009) estimated that direct consideration of costs during prioritization would have protected four times more distinct species and three times more threatened and endangered species than the observed allocation.

Several studies have found that cost variability across space or conservation actions may be at least as important, if not more important, than variability in environmental benefits (Polasky et al. 2001; Ferraro 2003; Bode et al. 2008). For example, in the previously mentioned analysis of a riparian buffer acquisition program, 92 percent of the benefits obtained by considering costs and benefits together could be gained by just considering costs; in contrast, recall that only 16 percent could be achieved by considering biophysical characteristics alone (Ferraro 2003). Similarly, Naidoo and Iwamura (2007) showed that cost-only targeting conserved biodiversity at lower costs than benefit-only targeting. When comparing different taxonomic groups as potential biodiversity surrogates across 34 global biodiversity hotspots, Bode et al. (2008) showed that conservation priorities were much more sensitive to variation in cost and the degree of threat facing a region than they were to changes in how biodiversity was measured.

Incorporation of costs is most important when costs and benefits are positively correlated, costs are more variable than biological measures, and budgets are low (Babcock et al. 1997; Ferraro 2003). The intuition behind this dependency is that cost-efficient targeting depends on the ratio of costs to benefits. If costs and benefits are negatively correlated, this ratio is large when benefits are large. However, when costs and benefits are positively correlated this ratio can be small even when benefits are large, so costs can change the prioritization of sites (Naidoo et al. 2006). Similarly, when costs are more variable than benefits, the variation in the ratios is more driven by the costs (Naidoo et al. 2006). Low budgets make cost-efficiency more important for meeting biological objectives. Bode et al. (2008) and Perhans et al. (2008) also found cost considerations to be particularly important in the absence of complementarity or target-based objectives (Carwardine 2010).

The small amount of existing empirical evidence points to a positive correlation between costs and conservation benefits (Naidoo et al. 2006). In addition, costs often vary by two to four orders of magnitude, while species richness or endemism typically vary by less than one (Naidoo et al 2006). For example, costs in the Ouachita Mountains on the North Atlantic Plain vary by more than an order of magnitude, while vertebrate species richness varies by less than 50 percent and plants by threefold (Murdoch et al. 2007). These explain the benefits of ROI analysis and underscore the importance of incorporating costs into conservation planning.

In contrast to previous reviews of the ROI literature (Hughey et al. 2003; Naidoo et al 2006; Murdoch et al 2007), we organize our synthesis around the way studies treat recurring, core elements of ROI, as a guide for practitioners and consumers of future ROI analyses. Our goal is to document the state of the art, note open questions, and provide suggestions for future improvements in data and methods. As described in Section 2, ROI analyses involve quantification of a consistent set of elements including the definition and measurement of the conservation objective as well as identification of the relevant baseline for comparison, the type of conservation investments evaluated, and measurement of investment costs. We discuss the literature's treatment of these distinct ROI components next.

### *4.1 The Biophysical Conservation Objective*

ROI-based conservation portfolio analysis maximizes a measure of conservation benefits for a given level of expenditures or minimizes the costs of achieving a predetermined conservation goal. The most commonly evaluated conservation benefit is some measure of biodiversity protection within a specified region. However, there is significant diversity in how this biophysical objective is evaluated and the scale at which actions are targeted.

Studies have been applied to diverse geographical scales, including global (e.g., Naidoo and Iwamura 2007; Carwardine et al. 2008a), transnational (e.g., Moore et al. 2004; Kark et al. 2009), ecoregional (Mandelik et al. 2010), biodiversity hotspots (Bode et al. 2008), specific countries or states (Ando et al. 1998; Carwardine et al. 2008b), and specific subregions of a country or state (Polasky et al. 2001; Arthur et al. 2004; Murdoch et al. 2010). The scale of individual conservation actions also has been diverse, with a focus on single land parcels (Ferraro 2003; Newburn et al. 2006), entire counties (Ando et al. 1998), or arbitrary land units of specific sizes (e.g., Carwardine et al. 2008b; Grantham et al. 2008).

Biodiversity measures can include genetic or taxonomic, species, or even ecosystem diversity. The majority of conservation ROI studies focus on protecting the greatest number of species. While measurement of total species richness is not feasible, most studies use single or multiple groups of species as surrogates. Specifically, conservation objectives include protection of the greatest number of vertebrates (Polasky et al. 2001; Arthur et al. 2004); vertebrate and plant species (Murdoch et al. 2007; Wilson et al. 2007; Underwood et al. 2008; White and Sadler 2011); threatened vertebrate and plant species (Murdoch et al. 2007); threatened vertebrate species and threatened endemic plants (Underwood et al. 2009); endangered plant and animal species (Ando et al. 1998); mammals (Carwardine et al. 2008a; Wilson et al. 2011); mammal and bird species (Williams et al. 2003); species threatened specifically by invasion (Evans et al. 2011); a specific taxonomic group, such as proteas (Grantham et al. 2008); endemic mammals, birds, reptiles, freshwater fishes, tiger beetles, terrestrial plants (Bode et al. 2008); reptiles, amphibians, and freshwater fish (Kark et al. 2009); vascular plants, beetles, moths, spiders, and small mammals (Mandelik et al. 2010); and vegetation types, environmental domains, and distributions of floral and faunal species of national significance (Carwardine et al. 2008b).

Some focus on measures of rarity, diversity, or taxonomic distinctiveness (Grantham et al. 2008; Joseph et al. 2009; Mandelik et al. 2010). Still others focus on less measurable aspects of biodiversity, such as threat reduction to coral reefs (Klein et al. 2010) or protection of "conservation features" that include specific biogeographic zones, habitat types, and species occurrences (Stewart and Possingham 2005).

Even with specific biophysical objectives in mind, there are many ways to link on-the-ground management to the measure of the benefit, and each approach makes different assumptions. Some studies assume that a species is protected if it occurs in at least one protected site (Ando et al. 1998; Polasky et al. 2001). If a species' presence at sites is uncertain, the objective alternatively can be to maximize the probability of occurrence (Arthur et al. 2004). A focus on species coverage, however, ignores issues of persistence that depend on ecological and evolutionary processes (Balmford et al. 2000) because species occurrence at a site does not guarantee persistence. In a study that takes a similar approach to species coverage, Murdoch et al. (2007) considered species-specific threats and assumed that a species will persist if its threat is abated. Newbold and Siikamäki (2008) used a spatially explicitly model of an endangered species' long-term persistence so that persistence can be maximized by the conservation program configuration.

In contrast, many studies have used habitat protection as a surrogate for species protection. Some set fixed targets (e.g., 10 or 15 percent) for the percentage of area, vegetation type, or species' historic range to protect (Stewart and Possingham 2005; Carwardine et al. 2008a; Carwardine et al. 2008b). Naidoo and Iwamura (2007) assumed that conserving 30 percent of an ecoregion conserves all its endemic species, and Balmford et al. (2000) assumed that a country's species are protected if 15 percent of its land area is protected. Wilson et al. (2011) set species-specific targets for different habitat types based on knowledge of each species, and Kark et al. (2009) set targets as percentages and total areas of species' range sizes, dependent in part on each species' current range and threat level.

Fixed targets imply threshold or linear benefits (Arponen et al. 2005; Wilson et al 2011). Goldstein et al. (2008) assumed constant returns to additional numbers of individual birds and plants protected by restoration, and Klein et al. (2010) focused on threat minimization, assuming that "threat" declines linearly with habitat protected.

Although a habitat area threshold may be appropriate for the preservation of particular species, the returns to habitat preservation generally diminish when the goal is to conserve multiple species (Davis et al. 2006; Wilson et al. 2006, 2009). For this reason, studies increasingly employ species-area curves (Rosenzweig 1995) to link land protection to species protection (e.g., Murdoch et al. 2007; Bode et al. 2008; Underwood et al. 2008; Underwood et al. 2009; Murdoch et al. 2010; White and Sadler 2011). The species area-curve links habitat area to the total number of species conserved, reflecting that additional species are protected at a diminishing rate with increasing total area protected. This same type of diminishing returns to protection has been used to link other types of conservation actions to species protection, such as the relationship between area of invasive species control and the number of species protected (e.g., Wilson et al. 2007; Evans et al. 2011).

Alternatively, benefits can account for species-specific conservation requirements. For example, Polasky et al (2008) considered species–habitat associations, species range, species–area requirements, and dispersal ability to predict the number of breeding pairs supported by different landscape configurations. They then use the number of breeding pairs to estimate the likelihood that each species is sustained on the landscape. Similarly, some studies incorporate reserve compactness or other such details into the benefit function by including connectivity bonuses (Murdoch et al. 2010).

A significant constraint to conservation ROI is identification of data on biophysical features and outcomes. Existing conservation ROI studies rely on data from a range of sources. Most employ previously compiled datasets. The publication of comprehensive, fine-resolution spatial databases of the world's mammals (Ceballos et al. 2005), birds (Orme et al. 2005), and amphibians (Stuart et al. 2004) has improved knowledge about patterns of species richness and endemism at the global scale (Lamoreux et al. 2006; Naidoo and Iwamura 2007). Ando et al. (1998) used county-level U.S. data on the estimated distribution of endangered species compiled by the U.S. Environmental Protection Agency. Balmford et al. (2000) employed mammal lists aggregated by country (Mace and Balmford 2000). Arthur et al. (2004) and Polasky et al. (2001) used data on the occurrence of terrestrial vertebrate species in Oregon, obtained from Master et al. (1995). Bode et al. (2008) used previously compiled data on the numbers of endemic mammals, birds, reptiles, freshwater fishes, tiger beetles, and terrestrial plants present in each terrestrial biodiversity hotspot (Mittermeier et al. 2004). Underwood et al. (2008) and Murdoch (2007) identified total numbers of plant and vertebrate species per ecoregion from existing databases (Kier et al. 2005; WWF 2006). Murdoch et al. (2007) were interested in the number of species at risk to specific threats, so they multiplied the total number of species in each region by the proportion of species on the IUCN red list for which each threat was listed as a major source of extinction risk. The Southeast Asia Mammal Database has the extent of occurrence and area of occupancy for more than 1,000 mammal species (Catullo et al. 2008); Wilson et al. (2011) used these data to classify habitat by its value to forest-dwelling mammals. Several studies (e.g., Carwardine et al. 2008a) have employed the global database of mammal distributions (Ceballos et al. 2006). Kark et al. (2009) used distribution range data compiled by the IUCN in the

Mediterranean Basin for amphibians, reptiles endemic to the Mediterranean countries, and endemic freshwater fish species. Stewart and Possingham (2005) used data on marine-related conservation features compiled from South Australian state government agencies. Carwardine et al. (2008b) employed a number of existing data layers to pinpoint vegetation types and environmental domains in Australia. They identified bird species' ranges by extrapolating from known occurrences. Evans et al. (2011) identified the spatial distribution of threatened species in Australia from a national database.

Some studies complement such data with field surveys to link conservation actions to biodiversity. For example, Mandelik et al. (2010) used field surveys to identify distributions of annual and perennial vascular plants, ground-dwelling beetles, moths, spiders, and small mammals. Smith et al. (2008) used a combination of remote sensing, ground truthing, and modeling to map land cover. Goldstein et al. (2008) employed field studies to determine the relationship between conservation targets (e.g., specific taxonomic groups or ecosystem services) and a land-use gradient for restoration.

Studies also have employed expert opinion, including to determine the probability of occurrence of each species at a site (Arthur et al. 2004), estimate the value of conservation easements for improving water quality at sites (Ferraro 2003), and parameterize how much a species' probability of survival would increase with a particular conservation project (Joseph et al. 2009).

Most studies that employ species-area relationships use a conventionally accepted functional form (Rosenzweig 1995), a shape parameter of 0.2, and total species richness specific to their focal region. White and Sadler (2011), however, used a functional form specifically developed for their region using field surveys. While parameterization of species-area relationships only requires information on the total number of species in a region, species lists are necessary for planners to account for complementarity of species across regions. Underwood et al. (2008) showed that doing so globally allowed three times more species to be protected than if complementarity were ignored.

As noted above, most studies focus on aggregate species measures and ignore species-specific habitat needs, such as area and quality. Two reasons are likely. First, species-specific needs are difficult to quantify and identify, requiring expert opinion. Second, greater complexity in the relationship between conservation actions and biophysical outcomes (such as including measures of compactness as well as total area) makes it more difficult to solve for the optimal conservation portfolio.

While biodiversity data are becoming increasingly available, acquiring and assembling data still can be challenging because datasets are not centrally warehoused and vary dramatically in their resolution and regional extent.

### *4.2 Predicting Baseline Conditions*

Three types of objectives have been the most common within the conservation ROI literature: maximizing the amount of biodiversity protected for a given budget, minimizing the cost of achieving a target level of biodiversity protection, and minimizing the loss of biodiversity over a given time period. These approaches make different implicit assumptions about the future of biodiversity in the absence of conservation actions—that is, the baseline.

Studies that select actions to maximize conservation benefits for a fixed budget implicitly assume that all unprotected areas or species will be lost or that risks are homogeneous across sites (Merenlender et al. 2009). Studies that focus on achieving conservation targets at the least cost (e.g., Stewart and Possingham 2005; Carwardine et al. 2008b; Kark et al. 2009) make the same assumption. In reality, biodiversity is likely to be maintained in areas that are not under direct protection by conservation actions. In fact, Wilson et al. (2011) found that failing to account for off-reserve biodiversity protection in East Kalimantan overestimated conservation costs by an order of magnitude.

An approach that seeks to minimize the loss of biodiversity over a specified time frame accounts for threats to biodiversity protection during the planning process (Murdoch et al. 2007; Bode et al. 2008; Grantham et al. 2008; Underwood et al. 2008). This approach employs a more realistic baseline because it does not assume that unprotected biodiversity is always lost. When species have some probability of surviving outside reserves, minimizing the loss of biodiversity is not the same as maximizing the number of species conserved in a reserve network (Witting and Loeschcke 1993, 1995).

Most studies account for the current baseline of protection and habitat loss by incorporating preexisting reserves into the eventual reserve network and excluding converted areas from analysis (e.g., Carwardine et al. 2008b; Underwood et al. 2008). Similarly, Evans et al. (2011) includes current invasive species control efforts when prioritizing future control.

As a general rule, approaches to maximize biodiversity protection tend to protect areas of low risk because costs and risks are correlated (Merenlender et al. 2009). In contrast, approaches to minimize biodiversity loss tend to justify investment in areas of moderate price, with moderate to high threat, that are biodiversity rich (Merenlender et al. 2009).

Threats are highly variable across space, and researchers can evaluate them using many approaches (Wilson et al. 2007; Wilson et al. 2009). Bode et al. (2008) estimated extinction risk for species based on the rate of species extinctions from IUCN Redbook listings and species-area curves. Klein et al. (2010) identified threats to coral reefs using existing data on the impact of anthropogenic drivers to change (Halpern et al. 2008). Studies have used the "Human Footprint" dataset (Sanderson et al. 2002) to predict intensity of threat across regions (e.g., Murdoch et al. 2007; Underwood et al. 2008; Murdoch et al. 2010). This dataset identifies areas of high human impact based on land use, population pressure, infrastructure and access but does not provide a direct measure of risk. Grantham et al. (2008) used previously observed rates of clearing to simulate spatially explicit, stochastic clearing of native vegetation in South Africa. Joseph et al. (2009) compared the likelihood of species survival with and without certain conservation actions, using expert opinion to determine survival probabilities. Klein et al. (2010) focused on reducing current threat levels, also based on expert opinion. Evans et al. (2011) explored species protection through invasive species control, assuming that the invasive species were the only threat and species would go extinct without control efforts; this study did not account for potential future spread of the invasive species.

While relatively rare, some studies model risk based on observed data. For example, Smith et al. (2008) predict patterns of land conversion based on distance from existing agriculture, elevation, slope and agricultural potential; however, they used threat as a measure of cost rather than represent threat itself. Newburn et al. (2005, 2006) estimate parcel-specific development probabilities using a model based on regional and site-specific characteristics. Similarly, Costello and Polasky (2004) predict land conversion threat based on projected urbanization using an existing model.

While prediction of threat and baseline conditions is fundamental to measurement of conservation's benefits, data on and inclusion of threats in ROI analysis is rare. Threat analysis—in particular analysis of threats beyond land conversion—are a future priority for ROI studies.

### *4.3 Types of Conservation Investment*

The vast majority of conservation ROI studies focus on land acquisition or creation of protected areas as the dominant conservation action. A few have focused on marine reserve creation (Stewart and Possingham 2005; Richardson et al. 2006), and Klein et al. (2010) focused on protection of both terrestrial and marine areas to reduce threats to coral reefs. Carwardine et al. (2008b) explicitly considered both acquisition and easements as potential control actions, while several others focused only on implementation of conservation easements (Ferraro 2003; Newburn et al. 2006). Polasky et al. (2008) prioritized land use zoning—designation of areas allowable for rural residential development, forestry, and agriculture—to achieve conservation benefits.

Researchers often use the creation of protected areas as an implicit surrogate for a broader range of conservation interventions (Margules and Pressey 2000; Chape et al. 2005; Wilson et al. 2009), including management of protected areas. Moore et al. (2004) selected protected areas, focusing on their management as the relevant action. Others have prioritized among specific management actions. Goldstein et al. (2008) selected vegetation types on which to focus restoration efforts. Wilson et al. (2011) and Low et al. (2010) considered a range of potential management changes to forested parcels. Evans et al. (2011) selected sites for rabbit or fox population control to reduce threats to native species. Joseph et al. (2009) prioritized among prespecified conservation projects for a variety of species. Wilson et al. (2007) prioritized a range of actions identified by expert opinion to abate threats to a variety of species. These actions included invasive riparian, predator, and weed control; fire management; and protection from urban and agricultural development.

### *4.4 How are Costs Measured?*

Cost measures are closely linked to the type of conservation investments explored. Cost measures can include the cost of acquiring land or easements, compensation that a landowner receives to improve or manage their land for conservation, transaction costs incurred to complete a conservation action, social costs borne by people disadvantaged by a project, and opportunity costs—profits or other benefits foregone by those participating in conservation. Some of these costs, including management, social, and opportunity costs, are incurred over multiple time periods, while others, like acquisition costs, are incurred immediately. Conservation ROI analyses have included different subsets of these costs and used different types and scales of data to quantify them.

Acquisition costs for land purchases have been estimated in a variety of ways. Frazee et al. (2003) and Pence et al. (2003) estimated acquisition and management costs in South Africa using expert knowledge and existing use values (Newburn et al. 2005). Murdoch et al. (2010) use an estimated purchase price as a proxy for conservation cost in Argentinean grasslands. They extrapolated per-hectare land costs for each parcel based on human population density and coarse land-cost maps created by an Argentine real estate company. Underwood et al. (2009) estimated land acquisition costs in each California county using public records on location, size, and price of conservation land acquisitions from 1990 to 2006. Ando et al. (1998) and Murdoch et al. (2007) estimated acquisition costs using U.S. county-level agricultural land values, a type of data that also exists for Europe (Naidoo et al. 2006). Ando et al. (1998) observed that the value of undeveloped land would have been a preferable proxy but that the agricultural land values reflects land market conditions. Polasky et al. (2001) and Arthur (2004) used the assessed market value of all nonurban land rather than just agricultural land to estimate the cost of protecting each private land site in their eastern Oregon focal areas. Carwardine et al. (2008b, 2010) used data on unimproved land values and sales cost provided by the Australian government, plus a transaction cost of US$10,000 for each site to represent administrative fees incurred when purchasing properties.

Another approach to estimating acquisition costs was based on a study of conservation management costs by Balmford et al. (2003), who found that per-area management costs are positively correlated with a county's purchasing power, economic output, and local human population density—and negatively related to the size of the protected area. They found that acquisition costs were about 50 times the magnitude of these management costs. Other studies have employed this relationship—and an improvement on it by Moore et al. (2004)—to estimate land acquisition costs (e.g., Bode et al. 2008; Underwood et al. 2008).

When land price coverage is incomplete, researchers can estimate more complete coverage of costs using statistical models (Naidoo et al. 2006). For example, to estimate acquisition costs in the Atlantic rainforest of Brazil, Chomitz et al. (2006) regressed land prices on characteristics such as soil type, climate, current land cover, and proximity to roads. Using the regression coefficients and geographic information system coverage of the relevant variables, they created a predicted land price layer for the entire study region.

Assessed land values tend to be more available in developed than developing countries (Naidoo et al. 2006). However, opportunity cost often can be used as a surrogate for acquisition costs because the value of land is tied to its long-term expected value from potential and current uses. Polasky et al. (2001) estimated the cost of conservation on public lands as the net present value of resource use, using forest inventory, site quality, and data on livestock forage productivity. Richardson et al. (2006) estimated opportunity costs of marine reserves using the first-sale value of fish and shellfish caught within large political units; they estimated opportunity costs at a finer resolution using interviews of fishermen. Stewart and Possingham (2005) estimated the costs of marine protected areas from the total catch of the southern rock lobster as reported for each planning unit. Polasky et al. (2008) considered biodiversity protection across a variety of land uses, including agriculture, forestry, and rural residential use, and they used models to predict the likely economic returns for each land parcel under the different land uses. Their approach used information on a parcel's soil, slope, elevation, and location to estimate yields, combined with commodity prices and production costs to generate economic returns for agriculture and forestry. They also estimated the value of rural residential development using information on location and characteristics of parcels.

The only global cost data currently available was developed by Naidoo and Iwamura (2007). This data, available at 5' (minute) resolution, estimates land costs as the flow of economic benefits derived from crops and livestock. They integrate spatial information on crop productivity, livestock density, and prices to produce a global map of the gross economic rents from agricultural lands. A number of ROI studies have used this data (e.g., Carwardine et al. 2008a).

Conservation on private land can sometimes be achieved without outright acquisition, by using conservation easements. Ferraro et al. (2003) estimated easement costs as 50 percent of land value plus a $5,000 transaction cost per parcel. Carwardine et al. (2008b) estimated these costs as the net present value of opportunity costs from agriculture based on estimates of price times yield minus production costs, plus the $10,000 transaction cost. Newburn et al. (2006) estimated easement costs in Sonoma County, California, by employing tax assessor records linked to a digital parcel map within a geographic information system that provided data on recent property sales, land use, and other site information. They estimated the value of development rights for each developable parcel as the value of developable land minus the restricted-use value, which they estimated using hedonic price models.

Another way to identify site-specific costs of conservation is the use of auctions. White and Sadler (2011) examined a perfect price-discriminating auction scheme, and Layton and Siikamäki (2009) used a reverse auction to elicit the willingness of nonindustrial private forest owners to preserve biodiversity-rich areas.

A variety of studies (e.g., Bode et al. 2008) have estimated management costs using the previously mentioned correlations developed by Balmford et al. (2003) and Moore et al. (2004). Kark et al. (2009) adapted this cost estimate to also include a measure of threat; their index equaled the product of human population density (used as a surrogate of threat to biodiversity) and management costs divided by the average population density over the Mediterranean Basin. Frazee et al. (2003) estimated the costs of management in the Cape Floristic Region based on expert opinion about what is needed to manage reserves of different sizes and the costs of those activities.

Wilson et al. (2007) focused on a diverse range of actions to abate threats to species and used expert opinion to estimate costs associated with each action. Evans et al. (2011) estimated the annual costs per unit area of fox and rabbit control in Australia from the literature and translated them into a single cost measure: the net present value of 10 years of management. Goldstein et al. (2008) estimated the costs of forest restoration activities based on existing literature and discussions with scientists, land managers, government employees, and other professionals knowledgeable about restoration in their focal region.

### *4.5 Additional Factors that Add Realism*

Few portfolio-based conservation projects can be fully implemented at the outset. For this reason, many conservation ROI studies describe a dynamic planning process (e.g., Murdoch et al. 2007; Wilson et al. 2007; Bode et al. 2008; Grantham et al. 2008; Underwood et al. 2008; Underwood et al. 2009; Evans et al. 2011). A dynamic planning approach allows adaptation during the implementation process, as decisionmakers can consider previous actions and the set of actions that are feasible during each phase of decisionmaking. This approach is particularly relevant when accounting for threats to species or sites and when focusing on minimizing short-term biodiversity loss (Bode et al. 2008; Grantham et al. 2008). A number of studies have explicitly considered development risk (Costello and Polasky 2004; Meir et al. 2004; Drechsler 2005; Haight et al. 2005; Strange et al. 2006).

Some studies also have considered the role of opportunism in conservation acquisitions. These studies explore the possibility that not all sites are immediately available for protection and that uncertainty exists about when and if a given site will be available (Meir et al. 2004; Drechsler 2005). Strange et al. (2006) also considered the possibility of selling parcels following acquisition, in response to either biodiversity loss on the parcel or changes in priorities due to the loss of availability of some unprotected parcels. But because none of these studies incorporate heterogeneous costs, decision rules have not been set forth that account for the combined influences of opportunism, risk, benefits, and costs.

An aspect of conservation planning not widely addressed in the existing literature is the endogeneity of land prices and threat in response to parcel conversion and protection (Costello and Polasky 2004). Because they provide amenity values, protected properties can drive up land prices and threats on neighboring parcels (Armsworth et al. 2006; Wilson et al. 2009). These price and threat dynamics could affect the timing and choice of conservation actions.

A diversity of other factors add additional realism to prioritizing conservation investments. For example, the success of different conservation actions is influenced by numerous sociopolitical factors, including political stability and corruption, budget continuity, governance, and stakeholder willingness to be involved in conservation initiatives (O'Connor et al. 2003; McBride et al. 2007; Wilson et al. 2009). The presence or persistence of biodiversity benefits across the landscape also may be uncertain (Polasky et al. 2000; McDonald-Madden et al. 2011). Some studies account for these uncertainties by modifying the expected biodiversity benefit (e.g., McBride et al. 2007; Joseph et al. 2009; Wilson et al. 2009).

While studies have begun to account for species complementarity when examining returns to land protection (Drechsler 2005; Underwood et al. 2008), complementarity across conservation actions has been largely ignored. For example, Wilson et al. (2007) assumed a species is protected via elimination of a single threat, without allowing for the possibility of multiple threats affecting a species. Evans et al. (2011) explicitly considered this type of complementarity. They point out that ignoring the issue of multiple threats can have two consequences: first, the benefits of abating a single threat may be overestimated because species may be threatened by multiple processes; second, the cost of abating two threats in one place may be cheaper than the sum of the costs of abating each threat alone.

Other factors that can affect conservation investment priorities include the potential that funds are not fully fungible (e.g., funds may need to be spent in specific areas), the potential for leveraging or partnering, and start-up costs associated with investing in a new region (Murdoch et al. 2007).

## 5. Multiobjective Conservation ROI

The preceding review makes clear that there is a substantial literature on conservation’s return on investment. To date, however, the empirical conservation literature has focused almost exclusively on single-objective, biodiversity-related ROI analysis. This is understandable because, as discussed in the previous section, single-objective ROI analysis is methodologically challenging in its own right, constrained by data availability, and beset by a rich set of still-unresolved issues. Multiobjective studies only add to ROI’s difficulty by expanding the set of outcomes and baselines to be modeled or measured. Moreover, multiobjective ROI studies reveal trade-offs between conservation outcomes.[2] If one project yields more biodiversity protection but less water quality improvement than another, which has the higher ROI? One of several ways to address the question is to translate outcomes into a common currency like dollars.

### *5.1 Conservation Missions and Ecosystem Service Objectives*

Despite the difficulties, changes in conservancies' missions are driving a practical need for multiobjective ROI tools. Increasingly, conservation organizations are expanding their missions to include social objectives. For example, Conservation International unveiled a new mission in 2010 to "empower societies to responsibly and sustainably care for nature *for the well-being of humanity*," and The Nature Conservancy's states its aim as "protecting nature, *for people today and future generations*" (emphases added).

Social missions imply measurement of conservation-related social outcomes, otherwise known as "ecosystem goods and services." Biodiversity-driven conservation can produce a broad range of ecosystem goods and services, including cleaner air and water, more productive soils, reduced risks of flooding and disease, aesthetic, cultural, and recreational benefits, and carbon storage. These types of benefits represent a suite of mission-driven returns to be evaluated.

### *5.2 Measuring Ecosystem Services' Biophysical Returns*

Consider the species-area relationships commonly used to predict the biodiversity benefits of land conservation. Ecosystem services analysis demands a suite of analogous empirical relationships that translate conservation investments into outcomes like improved water quality, water availability, carbon storage, soil quality, flood risks, human health risks, pollination services, and air quality. These relationships are referred to as "ecological production functions" (Daily and Matson 2008).

Empirical studies of these production functions are occurring across the vast and growing literature on ecosystem services. Examples include the relationship between land use or conservation and water quality (Baker 2003), marine species (Mumby et al. 2004), aquifer recharge (Scanlon et al. 2005), surface water availability (NRC 2008), pollination services (Ricketts 2004), and flood risk reduction (Mitsch and Day 2006). Of particular relevance are

[2] Note that ROI analysis is not the cause of these tradeoffs; it simply illuminates them.

modeling platforms being developed to assist planners with landscape-scale ecosystem service evaluation (Kareiva et al. 2011).

Despite these many efforts, ROI analysts lack empirically verified production relationships robust to a wide range of biophysical contexts. Put differently, most ecosystem services lack relationships like species-area curves that are deployable over broad conservation portfolios (Kremen 2005; U.S. EPA 2009). Notable exceptions include models that relate land cover to water quality (e.g., the SPARROW model) and the Universal Soil Loss Equation that relates land cover to erosion phenomena. Evaluating carbon storage as a function of land cover is another example of an empirically determined biophysical production function (e.g., Kindermann et al. 2008; Siikamäki and Newbold forthcoming). Causal relationships usually must be verified using rigorous, data-intensive empirical methods. Moreover, the nonuniformity of biophysical systems means that relationships verified in one context need not hold in others.

Of course, the same could be said of species-area curves. Species-area relationships are themselves not uniform, but enough empirical study over a wide range of contexts has created at least some degree of confidence in their use. In the near term, ROI analysis of ecosystem services will need to rely on production relationships that are qualitatively accurate and scientifically plausible but that may lack empirical validation. Validation and improved accuracy will take concerted empirical effort over a period of decades.

Another challenge for ecosystem services production analysis is that the beneficiaries and economic value of ecosystem services often are located far afield from the conservation action itself. Examples include down-watershed water quality availability and flood-related benefits arising from up-watershed land protections. Similarly, recreational, commercial, or subsistence benefits from habitat improvements in one location can be delivered across a species' entire migratory range. ROI analysis therefore requires ecological production functions that capture the spatial delivery of ecosystem services (Bockstael 1996; Boyd 2008).

### *5.3 Measuring Ecosystem Services' Economic Returns*

Predicted changes in ecological conditions via production function analysis do not by themselves capture the social returns to conservation. Assessment of social returns requires an additional step: translation of ecological returns into economic returns. The relationship between changes in natural systems and corresponding changes in economic welfare is a central focus of ecosystem services research (Heal 2000; U.S. EPA 2000; Boyd and Banzhaf 2007; Polasky 2008; Fisher et al. 2011). Ecological returns, such as improved soil productivity, more abundant species, flood and human health risk reductions, cleaner air and cleaner, more abundant water, are valuable. But how valuable?

Environmental and ecological economists over the last several decades have measured the economic value of a wide range of ecosystem goods and services, in specific spatial and social contexts (Freeman 2003). Boyd and Krupnick (2009) reviewed this literature, and the Environmental Valuation Reference Inventory provides a database of available studies. Valuation approaches include detection of environmental benefits in real estate prices (Polasky et al. 2011), referenda that approve local or state financing of land acquisitions (Banzhaf et al. 2010), inferences based on the travel and time costs people bear to enjoy natural resources (McConnell 1992), and experiments that ask people to make choices between money and environmental improvements in hypothetical settings (Carson et al. 2001). As a rule, the academic valuation literature finds clear evidence that ecological systems and the goods and services they produce are indeed economically valuable.

A challenge for measurement of economic returns is that the value of an ecosystem service improvement usually is highly dependent on the location where it is delivered (Boyd 2008; Polasky et al. 2008). As a rule, ROI analysts cannot simply apply the value of an ecological improvement detected in one location to another because the demand for ecosystem services is a function of the number of beneficiaries, the economic activities enhanced by the service, the availability of substitutes for the service, and other factors that depend on the characteristics of the social and biophysical landscape in question. The so-called benefit transfer literature addresses this problem. Benefit transfer methods take existing valuations derived from any of the aforementioned methods and transfer them to new landscape and resource contexts using statistical methods designed to control for similarities and differences in spatial context (Johnston 2007; Loomis and Rosenberger 2006).

### *5.4 Evaluatin Trade-Offs Across Multiple Objectives*

Trade-offs between ecosystem services arise when provision of one service comes at the cost of another. Synergies also can exist between ecosystem services, where the provision of one enhances the amount or quality of another (Raudsepp-Hearne et al. 2010).

Many ecosystem services depend on habitat protection and are therefore synergistic to some extent. But trade-offs regularly emerge. For example, the long-standing focus of ecosystem management on readily marketable ecosystem services, such as food, fiber, and timber, is commonly seen as a root cause behind the decline of other, less easily marketable ecosystem benefits, such as biodiversity or ecological functions and processes regulating the quantity and improving the quality of water.

Given that ecosystems provide multiple benefits, a focus on biodiversity as the only measure of benefit can result in suboptimal conservation decisions (unless biodiversity is all one cares about). For example, it is possible that a given conservation action can yield large water quality benefits and a high—but not maximum—biodiversity benefit. If water quality benefits are given no weight, conservation planning can fail to select the highest value projects.

Consider another pertinent example: conservation designed to achieve both biodiversity protection and carbon sequestration. In addition to providing potentially low-cost options for reducing global carbon emissions, reducing emissions from deforestation and forest degradation (REDD) projects could also support biodiversity conservation. Siikamäki and Newbold (forthcoming) examine the potential of global forest conservation programs to generate both biodiversity and carbon sequestration benefits. The global study of alternative forest conservation configurations indicates that the overlap between high-value sequestration and biodiversity areas is relatively small. The most attractive options for REDD programs occur where the opportunity cost per ton of avoided carbon dioxide emissions (accounting for the carbon content of the forest and the threat of deforestation) is lowest. Unfortunately, the correlation between species richness in major taxonomic groups and the opportunity cost per ton of avoided emissions is close to zero.

Given this type of result, how can an ROI analysis resolve the trade-off between sequestration and biodiversity? One approach is to convert biodiversity benefits into dollars so that that monetary benefit can be compared to the cost of more expensive forest sequestration options. Note that the reason economists favor translation of benefits into dollars is because monetary benefit measures simplify the comparison of "apples and oranges," like biodiversity and carbon sequestration.

When multiple outcomes are considered in an ROI framework, trade-offs can be evaluated, communicated, and resolved by weighting them via a common currency: i.e., dollars. Clarification of trade-offs is the reason environmental economists try to measure the monetary value of ecosystem services. The valuation methods described in the previous section are designed to measure the weight of different outcomes in a common currency. The problem, as noted earlier, is that the translation of many conservation outcomes—like biodiversity—can be difficult.

For this reason, studies like Polasky et al. (2005) and Nelson et al. (2009) are particularly relevant. These studies illuminate (but do not resolve) economic and biodiversity trade-offs by explicitly and formally depicting the biodiversity implications of planning that include multiple objectives, some of which are expressed in monetary terms.

In a study examining land use options and species conservation in the Willamette Basin in Oregon, Polasky et al. (2005) combined an economic model of agricultural production with an ecological model of species protection and use the model to identify spatially explicit land use options that maximize both economic and ecological objectives. Rather than monetizing ecological outcomes, they measured them using a species conservation score. They used the species conservation scores and monetary agricultural benefits to determine a "production possibilities frontier." This frontier depicts the maximum agricultural output value for any given level of species conservation benefit, thus illustrating the trade-offs involved in land use decisions. In their particular case, the analysis revealed that while increased species benefits comes at a cost of reduced agricultural output, the added cost is relatively small when activities are sited optimally. In general, the value of such analyses is that they can visually describe trade-offs without demanding the monetization of ecological outcomes. It is worth noting, however, that visual depiction of trade-offs becomes almost impossible when more than two objectives are being considered.

Nelson et al. (2009) also examined different land use options in Willamette Basin. The study evaluated several outcomes: agricultural production, soil conservation, carbon sequestration, water services, and species conservation outcomes. The study did not monetize ecosystem services but instead scored them in a variety of ways. In other words, the study did not "resolve" trade-offs by putting all outcomes in a common currency. However, the researchers described trade-offs in a manner similar to Polasky et al. (2005). The results suggest that scenarios that score high on ecosystem services also are beneficial to biodiversity. Perhaps not surprisingly, scenarios that target agricultural production had relatively high adverse impacts on ecosystem services and biodiversity.

This review of economic approaches to valuation and analysis of environmental trade-offs is far from exhaustive. We use it to emphasize that monetary valuation of ecosystem services is one way to derive the weights used in a multiobjective conservation ROI analysis. However, monetization is not strictly necessary if the goal is to describe, rather than resolve, trade-offs.

## 6. Conclusion

The academic literature provides ample evidence of ROI's value as a conservation planning tool. Widespread deployment of ROI analysis by conservancies will lead to better conservation outcomes and cost savings. Conservation ROI's future is its application to ecosystem services outcomes beyond biodiversity. This approach will not only help conservancies maximize conservation's broader benefits but also help foster stakeholder engagement and recruit new support by articulating conservation's diverse contributions to human well-being.

All organizations must make strategic choices about where to invest scarce resources and usually have to justify and communicate those choices to internal and external audiences. Because ROI analysis is an objective, analytically transparent, and data-driven approach to strategy and evaluation, it can help clarify the scientific and economic rationale for conservation.

Existing studies already challenge conservation managers to think differently by highlighting the importance of investment costs and clearly articulating assumptions regarding conservation's positive effect on biodiversity. The fact that ROI analysis tends to prescribe significant changes in conservation strategy (e.g., different portfolio selection and project location) is evidence enough of its value as a complement to existing planning tools.

Given ROI's promise and widespread acceptance in the academic conservation, natural resource management, and environmental economics literatures, why is it not deployed more in practice by conservation organizations? One reason is that data relevant to conservation ROI is relatively costly to collect. ROI analysis in the private sector can rely on a wide spectrum of market data—prices, sales, inventories, etc.—collected and reported by firms and governments in standardized formats. Environmental data is improving and can be applied to ROI analysis (as shown by this review of studies). However, ROI analysis clearly requires nontrivial investments in data, quantitative analysis, and human resources by conservation organizations

ROI's relevance and value as a tool also presumes that managers have the flexibility to choose from a wide array of investment possibilities. This may not always be the case. Conservancies with missions and activities constrained to smaller geographic regions, for example, have less investment flexibility, and thus less need for tools to help set priorities. Even global conservancies can face geographic constraints on investment imposed by limits on financing and administrative authority (e.g., a state program's need to spend its budget within its own borders).

ROI analysis also requires organizations to clearly articulate their objectives. In the private sector, this is usually straightforward and uncontroversial: profits and shareholder value are the goal. For nongovernmental conservation organizations, the articulation of objectives with enough specificity to foster quantitative measurement of outcomes is more difficult and controversial. This will be particularly true if and when conservancies seek to measure their social impact. As noted earlier, social outcomes, such as those associated with conservation's ecosystem services benefits, are multidimensional. Not only do they add to the evaluation burden, but they also trigger debate among stakeholders as to what an organization's specific objectives are and how they are to be weighted.

Conservation ROI's future lies in improved data and methods associated with assessment of threats (both biophysical and social) and experimental designs and models to measure or predict investment-related improvements in biodiversity and other ecosystem services. Further development of ROI will require a commitment on the part of conservancies to invest in data, methods, and skills—and to clearly articulate the ecological and social outcomes they most wish to pursue.

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